

DRAFT

**Environmental Implementation Guide
for Radiological Survey Procedures**

Office of Environmental Policy and Assistance
Assistant Secretary for Environment, Safety, and Health
U.S. Department of Energy
Washington, D.C. 20581

Draft Report for Comment
February 1997

Prepared by the
Measurement Application and Development Group
of the
Health Sciences Research Division
of
Oak Ridge National Laboratory

6. LABORATORY METHODS AND INSTRUMENTATION

6.1 INTRODUCTION

Samples collected during survey processes must be analyzed using the appropriate equipment and procedures. This manual assumes that the samples taken during the survey will be submitted primarily to a qualified laboratory for analysis. The laboratory must have written procedures that document its analytical capabilities for the radionuclides of interest and a QA/QC program that ensures the validity of the analytical results. The method used to assay for the nuclides of concern should be recognized as a factor affecting analysis time.

The most commonly used radiation detection and measuring equipment for radiological survey field applications has been described in Sect. 5. Many of these equipment types are also used for laboratory analyses, usually under more controlled conditions that provide for lower detection limits and greater delineation between radionuclides. Laboratory methods often involve combinations of both chemical and instrument techniques to quantify the low levels expected to be present in samples. This section provides guidance to assist the manual user in selecting appropriate procedures for specific applications. More detailed information is available in references provided in the reference section of this manual.

6.1.1 Prior Considerations

To reemphasize the point made in Sect. 2, a thorough knowledge of the radionuclides present, along with their chemical and physical forms and their relative abundance, is a prerequisite to selecting laboratory methods. With this information, it may be possible to substitute certain gross (i.e., non-radionuclide specific) measurement techniques for the more costly and time-consuming wet chemistry separation procedures, relating the gross data back to the relative quantities of specific contaminants. The individual responsible for the survey should be aware of the fact that chemical analyses of any samples require lead time that will vary according to the nature and complexity of the request. For example, a laboratory may provide fairly quick turnaround on gamma spectrometry analysis because computer-based systems are available for interpretation of gamma spectra. On the other hand, soil samples that must be dried and homogenized (and, in the case of ^{226}Ra , allowed to attain a known level of radon daughter ingrowth) require much longer lead time relative to samples that must not be dried (when, for instance, analysis for volatile chemicals or volatile radionuclides is desired). Some factors influencing the analysis time include (1) the nuclides of concern, (2) the number and type of samples to be analyzed, (3) the analytical method selected, (4) the QA/QC considerations required, (5) the availability of adequate equipment and personnel, and (6) the required detection limits.

6.1.2 Data Quality Objectives

Analytical methods should be capable of measuring levels below the established release guidelines: detection sensitivities of less than 10% of the guideline should be the target. Where costs, time, or other constraints make such sensitivities impracticable, higher sensitivities may be permitted. However, unless technically impracticable, methods selected should be capable of detecting 50% or less of the guideline value. Although laboratories will state detection limits,

these limits are usually based on ideal situations and may not be achievable under actual measurement conditions. Detection limits are subject to variation from sample to sample, instrument to instrument, and procedure to procedure, depending on sample size, geometry, background, instrument efficiency, chemical recovery, abundance of the radiations being measured, counting time, self-absorption in the prepared sample, and interferences from other radionuclides and/or materials present.

6.2 SAMPLE PREPARATION

Various degrees of sample preparation may be necessary prior to direct measurement and/or wet chemistry procedures. The only treatment for smears (filter papers) before gross alpha/beta counting will be to wait until short-lived naturally occurring radon daughters, which may have been collected along with the other radionuclides of concern, have decayed to negligible levels. For the ^{222}Rn and the ^{220}Rn series, decay times of 4 h and 72 h, respectively, are typically used.

6.2.1 Soil and Sediment

Soil and sediment sample preparation includes removal of sticks, vegetation, rocks exceeding about 0.6 cm ($\geq 1/4$ in.) in diameter, and foreign objects. If there is a possibility that a significant portion of the radioactive content of the sample may be associated with the larger size fraction, this portion of the sample should be analyzed separately to evaluate this distribution. If nonvolatile elements are the only contaminants of concern, the samples are dried at approximately 110°C for a minimum of 12 h; volatile radionuclides (^3H , ^{99}Tc , and iodides) must be separated from the sample before drying to avoid loss of the contaminant of interest. Dried samples are homogenized by mortar and pestle, jaw crusher, ball mill, parallel plate grinder, blender, or a combination of these techniques and sieved to obtain a uniform sample. Sieve sizes from 35 to 200 mesh are recommended for wet chemistry procedures. In addition, samples for chemical separations are usually ashed in a muffle furnace to remove any remaining organic materials that may interfere with the procedures. Care must be taken with certain elements, for example cesium, technetium, and zinc, which may volatilize at typical muffle furnace temperatures (i.e., approximately 450° C). Sample weights are determined as received and after drying and ashing procedures to enable referencing contamination levels back to weights of dry soil. To reduce the number of analyses required, multiple systematic or random samples from the same averaging region (i.e., equal aliquots from same grid block and same depth layer) may be combined into one composite sample. The number of samples combined into one composite must be limited to the maximum guideline concentration divided by the detection/measurement limit to ensure that the presence of one sample in excess of the guideline will be identified. The remainder of the individual samples should be retained to enable their analyses, in case composite sample analysis suggests the possibility of a hot-spot at one of the systematic or random sampling locations.

6.2.2 Water

Water samples are usually prepared by filtration of suspended material using a 0.45 micrometer filter and acidification with nitric or hydrochloric acid (or other appropriate acid) to a pH of less than 2. This permits separate analyses of suspended and dissolved fractions and, if preparation is not performed promptly following collection, prevents loss of dissolved

radionuclides by plating out on container surfaces (Berven et al., 1987). However, the possibility of volatilizing certain elements during acidification (e.g., technetium) must be considered when determining the appropriateness of the preparation step, and standard procedures should therefore be consulted. An additional precaution such as consulting the analysts for guidance prior to sampling is recommended.

6.3 ANALYTICAL PROCEDURES

This section briefly describes specific equipment and/or procedures to be used once the medium is prepared for analysis. The results of these analyses (i.e., the levels of radioactivity found in these samples) are the values used to determine the level of residual activity at a site. In the effort to release property for appropriate future use, the authorized limits for release are typically expressed in terms of the concentrations of certain nuclides. It is of vital importance, therefore, that the analyses be accurate and of adequate sensitivity for the nuclides of concern.

An excellent source of information on a variety of topics, from detection equipment to chemical procedures, is equipment vendor literature, catalogs, and instrument manuals. Other references that should be considered are available from such organizations as National Council on Radiation Protection and Measurements (NCRP), the U.S. Environmental Protection Agency (EPA), the American Society of Testing and Materials (ASTM), the DOE Technical Measurements Center (Grand Junction, CO), and the Environmental Measurements Laboratory (EML, formerly the Health and Safety Laboratory) of the DOE. Table 6.1 provides a summary of the common laboratory methods with estimated detection limits.

6.3.1 Analysis of Smears

- Gross Alpha/Gross Beta

The most popular method for laboratory smear and air filter analysis is to count both gross alpha and beta levels in a low-background proportional system; both automatic sample changer and manual multidetector instruments are used. Such systems have low backgrounds, relatively good detection sensitivity, and the capability of processing large quantities of samples in a short time. Using counting times of several minutes, measurement sensitivities of less than 10 dpm alpha and 20 dpm beta can be achieved. Filter papers can also be measured using standard field instruments, such as alpha scintillation and thin-window GM detectors with integrating scalers (see Sect. 5 on radiological detectors and instrumentation requirements). The measurement sensitivities of such techniques are not nearly as low as the low-background proportional system; however, for 5-min counting times, alpha and beta levels below 20 dpm and 100 dpm, respectively, can be measured. One of the major drawbacks to such a procedure is that it is very labor-intensive.

Filter papers can also be covered with a thin disk of zinc sulfide scintillator and counted for gross alpha using a photomultiplier tube attached to a scaler. While such a system provides a sensitivity comparable to that of the low-background proportional counter, it is also not usually automated and is, therefore, a labor-intensive method.

Table 6.1. Typical measurement sensitivities for laboratory radiometric procedures

Sample type	Radionuclides or radiation measured	Procedure	Approximate measurement sensitivity
Smears (filter paper)	Gross alpha	Low-background gas proportional counter; 5-min count.	5 dpm
		Alpha scintillation detector with scaler; 5-min count	20 dpm
	Gross beta	Low background gas proportional counter; 5-min count	10 dpm
		End window GM with scaler; 5-min count (unshielded detector)	80 dpm
	Low energy beta (^3H , ^{14}C , ^{63}Ni)	Liquid scintillation spectrometer; 5-min count	30 dpm
Soil sediment	^{137}Cs , ^{60}Co , ^{226}Ra (^{214}Bi) ^a , ^{232}Th (^{228}Ac) ^a , ^{235}U	Gamma spectrometry - Intrinsic germanium detector (25% relative efficiency); pulse height analyzer; 500-g sample; 15-min analysis	1– 3 pCi/g
	$^{234,235,238}\text{U}$; $^{238,239,240}\text{Pu}$; $^{228,230,232}\text{Th}$; other alpha emitters	Alpha spectrometry - pyrosulfate fusion and solvent extraction; surface barrier detector; pulse height analyzer; 1-g sample; 16-hr count	0.1–0.5 pCi/g
Water	Gross alpha	Low-background gas proportional counter; 100-ml sample, 200-min count	1 pCi/L
	Gross beta	Low-background gas proportional counter; 100-ml sample, 200-min count	1 pCi/L
	Miscellaneous gamma emitter	Gamma spectrometry - 3.5 L sample 16-hr count	10 pCi/L
	Miscellaneous alpha emitter	Alpha spectrometry - 100 ml sample; 16-hr count	0.1–0.5 pCi/L

^aIndicates number of progeny series measured to determine activity level of parent radionuclide of primary interest.

- Liquid Scintillation

Liquid scintillation is the preferred method for counting low-energy beta-emitters (for example ^3H , ^{14}C , and ^{63}Ni) and is excellent for counting high energy beta (^{32}P) and low-energy gamma emitters (^{125}I). Smears can be placed directly in a scintillation cocktail and counted on a liquid scintillation spectrometer. The counting efficiency may be reduced, but for the screening method, this process will yield reasonable results. With the spectrum capability of the newer instruments, the analyst can (in most cases) identify the specific beta emitter(s) present. The introduction of the sample into the liquid scintillation medium produces quenching, a reduction in the efficiency of the scintillator as a result of the introduction of the sample. To evaluate the effect of quenching, a known amount of the identified radionuclide (referred to as an internal standard or spike) may be added to the sample after initial measurement and a recount performed to enable determination of the detection efficiency of the specific sample. It should be noted that even with the identification of the nuclide(s) on the smears, this is still a gross analysis, and caution is advised in trying to infer too much from this information.

As a precaution against accidental contamination of the laboratory facility, it is prudent to first screen smears by gross GM or gamma counting. If little contamination is expected, all smears collected at the facility (or in a particular survey area) may be assayed at once by placing all the smears on the detector. This will provide a broad screen for expected and unexpected contaminants. If contamination is detected, the smears should be recounted in smaller groups until the contaminated smears are isolated. Since the procedure is nondestructive, it will not interfere with subsequent analysis of the smears. When performing such screening, the smears should be left in their protective “envelopes” to avoid cross contamination.

6.3.2 Analysis of Soil and Sediment

- Gamma Spectroscopy

After the soil or sediment has been prepared and placed in an appropriate container, the samples are counted. The analysis of soil or sediment is dependent on the radionuclides of interest. If the contaminants could include gamma emitters, the sample will be analyzed using gamma spectrometry (a nondestructive analysis that can identify and quantify multiple gamma-emitting nuclides). It is prudent to subject at least a representative number of soil or sediment samples to gamma spectral analysis, even if no gamma emitters are expected, as a check on the reliability of the identification of potential contaminants.

Either solid-state germanium detectors or sodium iodide scintillation detectors may be used; however, the solid-state detector has an advantage in its ability to resolve multiple gamma photopeaks that differ by as little as 0.5 to 1 keV of each other.

Although state-of-the-art systems include inherent computer-based spectrum analysis capabilities, it is important that an experienced analyst carefully review each spectrum because at the low concentrations typically encountered in radiological surveys, problems with resolution, interferences, peak shifts, and linearity may not be readily apparent. Spectra should also be reviewed for gamma photopeaks not previously identified as principal facility contaminants of concern. Special attention must be given to those radionuclides that may have difficult to resolve photopeaks, for example ^{226}Ra (186.2 keV) and ^{235}U (185.7 keV), and select, secondary photopeaks or daughter photopeaks for calculations. An example would be the use

of a daughter in the ^{226}Ra decay series, ^{214}Bi (609 keV peak), as an alternate for determining the quantity of ^{226}Ra present. When using such an approach, it is also necessary that the equilibrium status between the parent and daughters be known.

Soil/sediment analysis by gamma spectrometry can be performed with varying sample sizes, using geometries such as a 0.5 L Marinelli beaker, 100- to 400-mL cans or jars, various sizes of petri dishes, or standard 30-mL scintillation vials. Counting times ranging from one-half hour to 4 hours are usually adequate to detect most radionuclides at concentrations currently being used as cleanup guidelines. Longer counting times may be necessary for radionuclides with low gamma-emission rates (abundances) and/or low guideline concentration values.

- Alpha Spectroscopy (Chemical Separation)

Radionuclides emitting primarily alpha particles are best analyzed by wet chemistry separation, followed by counting to determine amounts of specific alpha energies present. Elements of concern can be removed from a solid sample by acid leaching, or samples can be fused at high temperatures into fluoride and pyrosulfate fluxes. This latter process ensures that all chemical species are in an ionic state that is more readily dissolved. (The process of leaching certain chemical forms of radionuclides from the soil matrix has been found to be less consistent than total dissolution of the sample matrix.) After dissolution, barium sulfate is precipitated to carry the alpha emitters out of solution. The precipitate is dissolved and the various nuclides are separated by oxidation-reduction reactions, or by ion exchange. After final separation and cleanup, the nuclides of interest are electroplated onto a metal disc or coprecipitated (with either neodymium or cerium fluoride) and collected on a filter paper. The metal disc or filter paper is then counted using a solid-state surface barrier detector and alpha spectrometer.

A known amount of tracer radionuclide is added to the sample before the chemical separation to determine the fraction of the radionuclide recovered in the procedure. This also provides a "calibration" of the analytical system for each sample processed. Lower limits of detection are less than 1 pCi/g using standard alpha spectrometry methods. Sample quantities for such procedures are typically a few grams or less.

- Other Procedures

Analysis of soil/sediment samples for most pure beta radionuclides, such as ^{90}Sr , ^{99}Tc , and ^{63}Ni requires wet chemistry separation, followed by counting using liquid scintillation or beta-proportional instruments. Each radionuclide (element) requires a specific procedure for the chemical separation; such detail is beyond the scope of this manual, and the reader should consult the references for further information. As with the alpha spectrometry techniques, a known amount of tracer is added to the sample to determine recovery. Lower limits of detection of less than 1 pCi/g are achievable using standard methods.

A recently introduced analytical technique uses liquid scintillation counting to measure alpha-emitting contaminant concentrations. This system is known as PERALS (photon electron rejecting alpha liquid scintillator). While this technique does not provide quite the resolution as conventional alpha spectrometry (solid-state detectors), it provides greater sensitivity, the chemical procedures are less rigorous, and the results are obtainable in a much shorter time (Perdue et al, 1978).

6.4 ANALYSIS OF WATER

Water samples may be directly counted for gamma emitters using the equipment described for soil/sediment samples. Because the guideline levels for appropriate future use are much lower for water than for soil (DOE 5400.5,) the larger sample volumes (1 to 3.5 L) and longer count times (up to 12 or 16 hours) may be necessary.

Gross alpha and gross beta analyses are conducted by evaporating a small (typically 10 to 100 mL) volume of water to dryness and counting on a low-background gas proportional system. Measurement sensitivities of 1 pCi/L are obtainable when low solids content limits self-absorption. Because of the substantial sample thickness that may occur, self-absorption may be significant and corrections will be required. Gross alpha/beta measurements are of low quality; the technique is intended primarily as a screening tool, and care must be used in interpreting data from these measurements. Samples that may contain radioactivity levels approaching guidelines should be analyzed further for specific radionuclides. Care must be exercised when the water may contain tritium, technetium, or other volatile radionuclides. In such circumstances, direct analyses by liquid scintillation or a combination of wet chemistry and liquid scintillation may be required. Analyses for other specific radionuclides are conducted in a manner similar to that for soil/sediment.

- Analysis of Tritium using Liquid Scintillation

If tritium is a radionuclide of concern, the tritium is separated by adding a known amount of low-tritium water and distilling the sample to collect the moisture. An aliquot of the collected moisture is then placed in a scintillation cocktail and counted using a liquid scintillation beta spectrometer. The activity is then related to the quantity of soil in the sample procedure or to the natural moisture content of the sample. Depending upon the moisture content of the sample and fraction disassociated by the distillation process, lower limits of detection on the order of several pCi/g can be obtained with this method. A technique for analyzing tritium in elemental form uses an oxidizer to convert tritium to water vapor that is collected in a cryogenic liquid bubble trap. An aliquot from the collecting trap is then placed in a scintillation cocktail and analyzed. Consult NCRP 57 for tritium measurement techniques (NCRP 1978).

7. INTERPRETATION OF SURVEY RESULTS

7.1 DATA CONVERSION

Radiological survey data are usually obtained in units such as counts per unit time that have no intrinsic meaning relative to the guideline values. Therefore, the survey data from field and laboratory measurements must be converted to units which will permit comparisons. Standard units used for expressing survey findings are:

- Surface contamination $\frac{\text{dpm}}{100 \text{ cm}^2}$ (disintegrations per minute per 100 cm²)
 $\frac{\text{Bq}}{\text{cm}^2}$ or
 (becquerels per cm²)
- Soil radionuclide $\frac{\text{pCi}}{\text{g}}$ (picocuries per gram) concentration
 $\frac{\text{Bq}}{\text{kg}}$ or
 (becquerels per kilogram)
- External exposure rate $\mu\text{R/h}$ (microroentgens per hour)
- Shallow dose rate mrad/h (millirads per hour)
 $\frac{\text{mGy}}{\text{h}}$ or
 (milligrays per hour)
- Dose equivalent rate $\frac{\mu\text{rem}}{\text{h}}$ (microrems per hour)
 $\frac{\mu\text{Sv}}{\text{h}}$ or
 (microsieverts per hour)

In performing the conversions it is necessary to know several factors; these are:

- c total integrated counts recorded by the measurement
- c/m gross count rate (counts per minute)
- t_g time period (minutes) over which the gross count was recorded
- t_B time period (minutes) over which the background count was recorded
- B count during recording period, due only to background levels of radiation
- B/m background count rate
- E detection efficiency of instrument in counts per disintegration
- A active surface area of the detector in cm²
- M mass of sample analyzed in grams (dry weight)
- 2.22 factor to convert a disintegration rate to activity units of picocuries, i.e., dpm/pCi.
- .0167 factor to convert dpm to Bq
- .037 factor to convert pCi to Bq
- .001 factor to convert g (grams) to kg (kilograms)
- .01 factor to convert mrad to mGy/h and $\mu\text{rem/h}$ to $\mu\text{Sv/h}$
- cf combination of all other factors needed for converting measurements in c/m to standard reporting units

These factors are used in the equations in the remainder of Sect. 7.1. All of Sect. 7.1 assumes that the cleanup guidelines for surface contamination are stated in units of dpm/100 cm².

7.1.1 Surface Activity

A measurement for surface activity is performed over an area represented by the sensitive surface area of the detector. If the measurement result is a count rate, i.e. in counts per minute, the conversion to dpm/100 cm² is performed by:

$$\frac{\text{dpm}}{100 \text{ cm}^2} = \frac{(c/m - B/m)}{E} \frac{100}{A} \quad (7.1A)$$

$$\text{or} \quad (7.1B)$$

$$\frac{\text{Bq}}{\text{cm}^2} = \frac{c/m - B/m}{E \cdot A} \cdot [0.0167]$$

For a technique using an integrated count on a digital instrument the conversion is:

$$\frac{\text{dpm}}{100 \text{ cm}^2} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E} \left[\frac{100}{A} \right] \quad (7.2A)$$

$$\text{or} \quad (7.2B)$$

$$\frac{\frac{B}{t_g}}{\text{cm}^2} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E \cdot A} (0.0167)$$

Care must be taken when calculating surface activity levels using a detector with a surface area differing from 100 cm². Generally, when the size of the contaminated region being measured is smaller than the probe area, the 100/A factor in Eqs. (7.1A) and (7.2A) should not be used. The 100/A correction factors are specifically included in these equations for measurements of areas that are larger than the probe size when the probe size is less than 100 cm². Probes with entrance window sizes greater than the maximum allowable averaging area used in the release criteria should not be used when making fixed point measurements.* For example, when evaluating the current 100-cm² surface contamination guideline, a probe area of around 100 cm² or less should be used. The use of a much larger probe size could result in an underestimate of activity within a small 100-cm² area beneath the probe since the remaining, less contaminated region surrounding the high activity spot will reduce the response of the detector. The response will accurately reflect the average contamination beneath the probe, but the smaller activity area, which may exceed the 100-cm² release limit, may go unnoticed.

*Probe sizes larger than 100 cm² are recommended for use during scan surveys when the detection sensitivity is adequate.

The level of removable activity collected by a smear is calculated generally in the same manner as for direct measurements, except, because the smear itself is performed over a 100-cm² area and the detector geometry correction is not considered when determining the efficiency, the detector area correction factor is not necessary for Eq. (7.2A). The equations for calculating removable activity are:

$$\frac{\text{dpm}}{100 \text{ cm}^2} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E} \quad (7.3A)$$

or

$$\frac{\text{Bq}}{\text{cm}^2} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E \cdot 100} \cdot [0.0167] \quad (7.3B)$$

- Surface Dose Rate

A beta surface dose rate can be determined by taking a measurement with a beta-sensitive detector as long as the appropriate conversion factor is used. If the instrument display is in count rate (*c/m*), the conversion to mrad/h is given by:

$$\text{mrad/h} = (c/m - B/m) (cf) \quad (7.4A)$$

or

$$\text{mGy/h} = (c/m - B/m) (cf) (0.01) \quad (7.4B)$$

The conversion factor (*cf*) will vary for the different beta energies given by different isotopes. To determine a surface dose rate, the specific contaminant(s) and response of the detector being used must be known.

7.1.2 Soil Radionuclide Concentration

To determine the radionuclide concentration in soil in units of pCi/g (dry weight) the calculation performed is:

$$\text{pCi/g} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E \cdot 2.22 \cdot M} \quad (7.5A)$$

or

$$\frac{\text{Bq}}{\text{kg}} = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{t \cdot E \cdot M} \cdot \left[0.0167 \cdot \frac{1}{0.001} \right] \quad (7.5B)$$

It should be noted that concentrations in soil are to be expressed in terms of dry weight, and the soil will either require drying before analyses or a correction factor for moisture content applied in the calculation. If the analytical procedure includes a wet chemistry separation, it will also be necessary to correct for the fractional recovery (R) determined by a spike or tracer added to the sample.

$$pCi/g = \frac{\frac{c}{t_g} - \frac{B}{t_B}}{E \cdot 2.22 \cdot M \cdot R} \quad (7.6)$$

7.1.3 External Exposure Rate and Dose Equivalent Rate

If an instrument such as a pressurized ionization chamber is used for measuring exposure rate, the instrument reading will be in the desired exposure rate units. Data in units of counts per minute or per some preset time can be obtained by combining either a gamma scintillation detector or a GM detector with one of two instruments; i.e., a count rate instrument or a digital scaling instrument. Conversion to exposure rate is accomplished using calibration factors developed for the specific instrument and survey site. It is possible that release criteria will be expressed in terms of dose equivalent rate (μ rem/h or μ Sv/h). Tissue equivalent detectors are available which allow direct measurement of dose equivalent rates, but at the time of this writing these instruments are generally not very stable in the 10 to 20 μ rem/h range. For purposes of measuring most environmental levels, one can assume that a direct gamma exposure of 1 μ R is equivalent to 1 μ rem. Given this assumption, pressurized ionization chamber measurements can be directly applied to dose equivalent rate comparisons. This net level is compared with the guideline value (cf is the site-specific calibration factor for the detector).

$$\mu R/h = (c/m - B/m) \cdot cf \quad (7.7A)$$

$$\mu Sv/h = c/m \cdot cf \cdot 0.01 \quad (7.7B)$$

$$\mu rem/h = c/m \cdot cf \quad (7.7C)$$

7.2 MEASUREMENT UNCERTAINTY*

The quality of measurement data will be directly impacted by the magnitude of the uncertainty associated with it. Some uncertainties, such as statistical counting uncertainties, can be easily calculated by mathematical procedure directly from the count results. Evaluation of other sources of uncertainty require more effort and in some cases is not possible. For example, if an alpha measurement is made on a porous concrete surface, the observed instrument response when converted to units of activity will probably not equal the true activity under the probe. Variations in the absorption

* Throughout Section 7.2, the term measurement uncertainty is used interchangeably with the term standard deviation. In this respect, the uncertainty is being qualified as being numerically identical to the standard deviation associated with a normally distributed range of values.

properties of the surface for particulate radiation will vary from point to point and therefore will create some level of variation in the expected detection efficiency. The analysis of uncertainty, as discussed in this section, should be applied in a reasonable fashion. The level of analysis should match the need and usefulness for the expected use of the data.

7.2.1 Systematic and Random Uncertainties

Measurement uncertainties are often broken into two sub-classes of uncertainty and termed systematic (i.e., methodical) uncertainty and random (i.e., stochastic) uncertainty. Systematic uncertainties derive from lack of knowledge about the true distribution of values associated with a numerical parameter. An example of a systematic uncertainty would be the use of a single counting efficiency for a given gamma energy when counting soil samples. The person performing the measurements has a judgmental confidence that the value will probably be a little different each time because the atomic components of the sample media will be different each time. He doesn't really know what the value is for any particular measurement or the true distribution of values, but he can make a reasonable guess at an upper and a lower limit. It would be unreasonable (i. e., cost prohibitive) to actually attempt to account for a variation such as this for each sample unless the estimated uncertainty was large relative to the values being obtained. Random uncertainties refer to fluctuations associated with a known or expected distribution of values. An example of a random uncertainty would be a well-documented chemical separation efficiency which is known to fluctuate with a regular pattern about a mean. A constant recovery value is used during calculations, but the true value is known to fluctuate from sample to sample with a fixed and known degree of variation. A certain amount of uncertainty is expected in the final value and the degree of uncertainty is relatively well understood. A third type of uncertainty, for lack of a better term, is called a mistake. Miscalculations and badly assumed values account for many mistakes in both the data collection and reduction phases of research and analysis. The only true way to detect and fix uncertainties of this type is through validation and peer review.

To limit the need for estimating potential sources of uncertainty, the sources of uncertainty themselves should be reduced to a minimum level by using the following practices.

- The detector used should minimize the potential uncertainty. For example, when making field surface activity measurements for ^{238}U on concrete, a beta detector such as a Geiger-Mueller pancake may provide better quality data than an alpha detector, depending on the circumstances. Less random uncertainty would be expected between measurements with a beta detector such as a pancake, because beta emissions from the uranium will be affected much less by thin absorbent layers than will the alpha emissions. Refer to Sect. 4 for discussions pertaining to the selection and use of instrumentation.
- Uncertainties should be either reduced or eliminated by use of standardized measurement protocols when possible. Special effort should be made to reduce

systematic uncertainties when the magnitude of such variations is significant relative to the final quantity of interest. Effective peer review will be a necessary part of this effort.

- Professional judgement should be used when considering the need for uncertainty analysis. For most actions involved with radiological surveys, complete uncertainty analyses should only be considered for controllable measurements such as laboratory sample analysis. The number of factors that affect field measurements would require an inordinate amount of time to evaluate and the benefit will not usually be expected to outweigh the cost. This is not to say that factors that affect field measurements should be ignored. Refer to Sect. 4.3, Radiation Measurements, for a discussion of factors which will adversely affect field measurements with portable instrumentation and how to properly consider these effects.

For uncertainties that cannot be eliminated, the independent effects can be propagated into the final data as described in Sect. 7.2.3. As stated above, non-statistical uncertainties should be minimized as much as reasonably possible through the use of good work practices, proper calibrations, and effective peer review.

7.2.2 Statistical Counting Uncertainty

When performing an analysis with a radiation detector, the result will have an uncertainty associated with it due to the statistical nature of radioactive decay. To calculate the total uncertainty associated with the counting process, both the background and the sample measurement uncertainties must be accounted for. The standard deviation of the net count rate, or the statistical counting uncertainty, can be calculated by

$$\sigma_n = \sqrt{\frac{N_g}{t_g^2} + \frac{N_b}{t_b^2}} \quad (7.8)$$

where

- σ_n = standard deviation of the net count rate result
- N_g = number of gross counts (sample)
- t_g = gross count time
- N_b = number of background counts
- t_b = background count time

7.2.3 Uncertainty Propagation

Most measurement data will be converted to different units or otherwise included in a calculation to determine a final result. The standard deviation associated with the final result, or the total uncertainty, can then be calculated. Assuming that the individual uncertainties are relatively small, symmetric about zero, and independent of one another, then the total uncertainty for the final calculated result can be determined by solution of the following partial differential equation (Knoll 1979):

$$\sigma_u = \sqrt{\frac{Mu^2}{Mx} \sigma_x^2 + \frac{Mu^2}{My} \sigma_y^2 + \frac{Mu^2}{Mz} \sigma_z^2 + \dots} \quad (7.9)$$

where

- u = function, or formula, that defines the calculation of a final result as a function of the collected data. All variables in this equation, i.e., x, y, z, \dots , are assumed to have a measurement uncertainty associated with them and do not include numerical constants
- σ_u = standard deviation, or uncertainty, associated with the final result
- $\sigma_x, \sigma_y, \dots$ = standard deviation, or uncertainty, associated with the parameters x, y, z, \dots

Equation (7.9), generally known as the error propagation formula, can be solved to determine the standard deviation of a final result from calculations involving measurement data and their associated uncertainties. Recognizing that most users of this manual will not be comfortable with the manipulation of differential equations, the solutions for common calculations are included below.

<u>Data Calculation</u> $u = x + y$, or $u = x - y$:	<u>Uncertainty Propagation</u> $\sigma_u = \sqrt{\sigma_x^2 + \sigma_y^2}$	(7.10A)
---	---	---------

$u = x \div y$, or $u = x \cdot y$:	$\sigma_u = u \sqrt{\frac{\sigma_x^2}{x^2} + \frac{\sigma_y^2}{y^2}}$	(7.10B)
---------------------------------------	---	---------

$u = c \cdot x$, where c = constant:	$\sigma_u = c \sigma_x$	(7.10C)
---	-------------------------	---------

$u = x \div c$, where c = constant:	$\sigma_u = \frac{\sigma_x}{c}$	(7.10D)
--	---------------------------------	---------

Note: In the above solutions, x and y are measurement values with associated standard deviations, or uncertainties, equal to σ_x and σ_y , respectively. The symbol "c" is used to represent a numerical constant which has no associated uncertainty. The symbol σ_u is used to denote the standard deviation, or uncertainty, of the final calculated value u .

7.2.4 Reporting Confidence Intervals

Throughout Sect. 7.2, the term measurement uncertainty has been used interchangeably with the term standard deviation. In this respect, the uncertainty is being qualified as being numerically identical to the standard deviation associated with a normally distributed range of values. When reporting a confidence interval for a value, one provides the range of values that represent a predetermined confidence level. To make this calculation, the final standard deviation, or total uncertainty σ_u as shown in Eq. (7.9), is multiplied by a constant factor k representing the area under a normal curve.

The values of k representing various intervals about a mean of a normal distribution as a function of the standard deviation is given in Table 7.1. The following example illustrates the use of this factor in context with the propagation and reporting of uncertainty values.

Table 7.1. Areas under various intervals about the mean, $\bar{\mu}$, of a normal distribution with a standard deviation σ

Interval ($\bar{\mu} \pm k\sigma$)	Area
$\bar{\mu} \pm 0.674\sigma$	0.500
$\bar{\mu} \pm 1.00\sigma$	0.683
$\bar{\mu} \pm 1.65\sigma$	0.900
$\bar{\mu} \pm 1.96\sigma$	0.950
$\bar{\mu} \pm 2.00\sigma$	0.954
$\bar{\mu} \pm 2.58\sigma$	0.990
$\bar{\mu} \pm 3.00\sigma$	0.997

Example: Uncertainty Propagation and Confidence Interval

A measurement process with a background of 1 count in 10 minutes yields a sample count result of 28 ± 5 counts in 5 minutes, where the ± 5 counts represents one standard deviation about a mean value of 28 counts. The detection efficiency is 0.1 counts per disintegration ± 0.03 counts per disintegration, again representing one standard deviation about the mean.

Calculate the activity of the sample, in dpm, total measurement uncertainty, and the 95% confidence interval for the result.

1. The net count rate is:

$$\frac{28 \text{ counts}}{5 \text{ min}} - \frac{1 \text{ count}}{10 \text{ min}} = 5.5 \text{ cpm}$$

2. The net count rate uncertainty is: [Eq. (7.8)]

$$\sqrt{\frac{28}{5^2} + \frac{1}{10^2}} = 1 \text{ cpm}$$

3. The activity is:

$$\frac{5.5 \text{ cpm}}{0.1 \text{ cpm / dpm}} = 55 \text{ dpm}$$

4. The uncertainty for the activity is: [Eq. (7.10B)] for division in step 3)

$$55 \text{ dpm} \sqrt{\frac{1^2}{5.5} + \frac{0.03^2}{0.1}} = 19 \text{ dpm}$$

5. The activity will then be 55 dpm and the total uncertainty will be 19 dpm. (Since the count time is considered to have trivial variance, then it is assumed to be a constant.)
6. Referring to Table 7.1, a k value of ± 1.96 represents a confidence interval equal to 95% about the mean of a normal distribution. Therefore, the 95% confidence interval would be $1.96 \times 19 \text{ dpm} = 37 \text{ dpm}$. The final result is $56 \pm 37 \text{ dpm}$.

Please note that the uncertainty just calculated only represents the uncertainty associated with the analysis of the sample and does not include any potential errors or uncertainties associated with the collection of the sample or with the assumptions made about the representation of the media being sampled.

7.3 DETECTION SENSITIVITIES

The concept of detection sensitivities was introduced and discussed in Sect. 5. For the purposes of thorough data presentation, the detection sensitivity for each measurement procedure (and each instrument if more than one instrument is used for a given procedure) is calculated. Data from surveys will often be near background levels and/or may not be discernable from background. Many measurements near background levels may be at or below the critical detection level of the measurement equipment being used. All measurements above the critical level should be considered positive and reported with a 2σ error as discussed in Sect. 7.2. Measurements below the critical level, L_c , should be considered as background for comparison with guideline levels. However, if any measurement below the mean background is to be included in an averaging process to determine a mean value for an area or unit, the value should be used as measured, i.e., both positive and negative values should be used. Negative data will be a frequent result of calculations. Use of the MDA or critical level for data that have a value less than the detection capability is a common practice accepted by the EPA (EPA 1989b). This approach enables the surveyor to significantly reduce the number of calculations; however, use of a detection capability in place of actual data when calculating averages will bias the results on the high side, and the true conditions of the site will not be described. Substituting detection sensitivities for actual data will also result in overestimates of source inventory and dose assessments, possibly leading to decisions for further actions that may not be justified. Finally, when evaluating data distributions (e.g. in a normalcy test), use of detection sensitivity values will result in a skewed distribution and may lead to incorrect conclusions. To avoid the pitfalls associated with use of detection sensitivity values, it is recommended that actual data be presented and used for calculational purposes. One exception to this approach might be the use of such values for averaging site activity levels when the detection sensitivity

is small in comparison to the applicable guideline. For the purposes of this Manual, small may be considered as less than 10% of the guideline value.

7.4 FORMAT FOR DATA PRESENTATION

All analytical data from the surveys should be presented in a format which provides (1) the calculated surface activity or specific radionuclide concentration value; (2) the estimated uncertainty at the 95% confidence level for that value; and (3) the estimated detection sensitivity (MDA and/or critical levels) for the measurement (EPA 1980a). When reporting less-than-detectable data, actual numerical values should be included [DOE/EH-173T (1991a)]. Note that in the following examples of acceptable formats, the critical detection level is given in units of activity. The formulas presented in Sect. 5 for the calculation of Critical Detection Levels assume that the end result will be in units of counts. However, when reporting any value, it is recommended that the final result be presented in a unit of activity that is consistent with the unit of the cleanup guideline.

In the following example,

- the critical detection level for direct alpha measurements was 25 dpm/100 cm²,
- the MDA for direct alpha measurements was 60 dpm/100 cm²,
- the critical detection level for direct beta measurements was 420 dpm/100 cm²,
- the MDA for direct beta measurements was 950 dpm/100 cm².

Actual value available for "Not Detected" results (2 significant digits)

Location ID	Directly measured activity levels ^a	
	Alpha (dpm/100 cm ²)	Beta (dpm/100 cm ²)
A1	[7]	[-30]
A2	35	460
A3	[0]	[100]
A4	58	620

^aAll values represent the actual measurement less the background response of the detector used. A value in brackets [##] indicates that the measurement was not discernable from the background response of the detector (95% confidence level).

The table below is an example of reporting analytical results when values below the detection limit are not available. Many commercial analysis packages are designed to produce only the specific detection limit when the calculated result is "Not Detected." Under this condition, the following format provides an example for documenting the results. When possible, an actual value should be obtained and reported as shown in the previous example.

Actual value NOT available for “Not Detected” results
(2 significant digits)

Location I.D.	Radionuclide concentration ^a	
	²³⁸ U	¹³⁷ Cs
A1	2.2 ± 0.1	<0.8
A2	1.5 ± 0.2	<1.5
A3	1.8 ± 0.1	2.3 ± 0.6
A4	38 ± 2	3.5 ± 0.5

^aAll values represent the actual measurement less the background response of the detector used. A less-than symbol (<) indicates that the measurement was not discernable from the background response of the detector (95% confidence level). The value following the < symbol is the critical detection limit for the indicated radionuclide during the sample count.

In expressing survey results, the number of significant figures is also of importance. Data reported should be reasonable and not mislead or imply a false level of accuracy. The appropriate number of digits in a value depends upon the magnitude of the uncertainty attached to that value. In general, final survey data, which are usually in the range of environmental data, seldom justify more than two or three significant figures for the value and one or two significant figures for the uncertainty of the surface activity measurement. When reported, actual values should be listed in parentheses following the detectable limit [e.g., <450 (-120)].

7.5 DIRECT COMPARISON OF SURVEY UNIT RESULTS WITH GUIDELINE VALUES

This section provides guidance on calculations required when performing direct comparisons of survey results to the current generic release guidelines and site-specific derived concentration guidelines presented in Appendix A. When the survey average and weighted average results are obviously less than the guideline value for any given grid block, then no further action is needed other than the direct comparisons as discussed here.

If the results from a contiguous group of survey units indicate that the remaining activity level within the grouped region is near the release limit, i.e. within 20%, then additional evaluations may be warranted. For general application, if the activity level is 80% or more of the release guideline within the following sized areas then additional evaluations may be necessary: (1) an indoor area covering an entire room or more than 10 m², or (2) an outdoor area covering an entire survey site or more than 1000 m². The collected data and calculations performed should be reviewed with the intent of deciding whether or not enough data have been collected to support the decision that the remaining activity is truly below the release guideline. There are no prescriptive rules for making these types of decisions—professional judgement is required and knowledge of the measurement and analysis methods is essential.

7.5.1 Calculating Average Levels

General surface activity, soil activity, and exposure rate guideline values are average values, above background, established for areas of survey unit surfaces (surface activity), 100 m² (soil activity), and open land (exposure rates). To allow comparison of the survey data with those guidelines, the mean (\bar{x}) of measurements in each of the survey units (or grid areas) is calculated using all measurements (n_s) within that area:

$$\bar{x} = \frac{1}{n_s} \sum_{i=1}^{n_s} x_i \quad (7.11)$$

7.5.2 Removable Activity

Release guidelines for transferable contamination are typically stated in terms of activity per unit mass or per unit area, averaged over an area; e.g., pCi/g averaged over 100 m² or dpm/100 cm² averaged over a specified surface area. Data for removable activity levels are compared directly to the guideline values. The limit for removable activity is 20% of the guideline value for total surface activity (Order DOE 5480.6, 1986). Any result in excess of that level must be addressed, and the area may require remediation.

7.5.3 Areas of Elevated Activity

Levels of residual activity (i.e., elevated areas) that exceed the guideline value are initially compared directly with the guideline.

- Buildings or Structures

The limit for activity on a building or structure surface is three times the guideline value when averaged over a single 100-cm² area. Residual activity exceeding this limit must be remediated and followup surveys performed. Areas of elevated activity between one and three times the guideline value are then tested to ensure that the average surface activity level within a contiguous 1-m² area containing the elevated area is less than the guideline value.

To evaluate whether this averaging condition is satisfied, additional measurements are performed, and the activity level and areal extent of the elevated area are determined. The average (weighted average) in the 1-m² area is then calculated, taking into consideration the relative fraction of the 1 m² occupied by the elevated area(s), using the relationship:

$$x_w = \sum_{i=1}^n \left[A_i \frac{x_i}{T} \right] \quad (7.12)$$

where

x_w	=	weighted average [including elevated area(s)],
A_i	=	average activity in area i (dpm/100 cm ²),
x_i	=	areal size of area i (cm ²),

T = total areal size of region being evaluated,
 n = number of areas [including non-elevated area surrounding the hot spots].

Sample Calculation

The survey has identified an area of surface activity having an average level of 7000 dpm/100 cm² and occupying an area of 800 cm². Five measurements in the contiguous 1 m², outside the elevated area, are each less than the guideline value of 5000 dpm/100 cm², average 2300 dpm/100 cm², and occupy an area equal to (10,000 cm² - 800 cm²) 9200 cm². The weighted mean for the 1-m² (10,000 cm²) area containing the elevated area is

$$7000 \left[\frac{800}{10000} \right] + 2300 \left[\frac{9200}{10000} \right] = 2676 \text{ dpm / } 100 \text{ cm}^2$$

• Soil

Areas of elevated activity between one and three times the guideline value are tested to ensure that the average concentration is less than (100/A)^{1/2} times the guideline value, where A is the area of the elevated activity in m². Levels exceeding this limit must be remediated. If this condition is satisfied, the average activity in the 100-m² contiguous area containing the region of elevated radionuclides is then determined to ensure that it is within the guideline value. Equation (7.12) is also used for this calculation, substituting 100 m² for the 1 m² used when calculating average surface activity.

Sample Calculation

Five systematic soil samples from a 100-m² grid block have the following concentrations of a specific radionuclide resulting in an average concentration value of 2.9 pCi/g:

1.5 pCi/g	1.6 pCi/g
2.7 pCi/g	3.5 pCi/g
5.0 pCi/g	

In addition, this grid block contains a 20-m² elevated area with an average soil concentration of 15.5 pCi/g. Using the relationship of (100/A)^{1/2}, the 20-m² area would be permitted to have an average concentration of (100/20)^{1/2} or 2.236 times the guideline value; i.e., for a guideline of 10 pCi/g, this value becomes 22.36 pCi/g. The activity level of 15.5 pCi/g in this elevated area satisfied this limit. The weighted average for the contiguous 100 m² containing the elevated area is

$$\begin{aligned} \bar{x}_w &= 2.9 \left[\frac{80}{100} \right] + 15.5 \left[\frac{20}{100} \right] \\ &= 2.32 + 3.10 \\ &= 5.42 \text{ pCi/g.} \end{aligned}$$

7.5.4 Exposure Rates

Exposure rate levels are compared directly with the guideline value. The maximum exposure rate may not exceed 20 μ R/h above background in any habitable structure [DOE/CH/8901 (DOE 1989a)]. If the level is above that value, the area must be remediated and resurveyed.

7.6 STATISTICAL TESTS

Section 7.5 discusses the direct comparisons that must be made when evaluating survey data within the context of the current DOE cleanup guideline structure. As mentioned in Sect. 1.3, alternate dose-based cleanup guidelines that are intended for demonstrating compliance across large survey units may be developed and approved by the Department.* As such, statistical evaluations will often be necessary for proving compliance with the release criterion.

Even when using a large area dose-based cleanup approach, the amount of activity that can be allowed to remain within small elevated areas will be limited. It will be possible for small grid blocks within a survey unit to exceed the large-area, derived cleanup guide yet not exceed any small-area cleanup guides. The goal of evaluating the resulting data is to determine whether the average contamination level within the survey unit meets the cleanup guide with an *a priori* level of confidence. The confidence level is a measure of the *expected* variability of the true contamination levels based on a group of independent data values, each of which is assumed to represent the average contamination level within a sub-region of the survey unit.

Three different statistical tests are presented in this section: (1) Comparisons when data are normally distributed and the background is a known constant, (2) Non-parametric upper 95% confidence limit test, and (3) Comparing survey unit data with background data. Although these methods are presented here and are believed to cover the majority of cases that will be encountered during radiological surveys, it is not an exhaustive listing and therefore should not be interpreted to mean that other methods are any less valid. Prior to discussing the statistical tests, an overview of how to formulate data sets from sample results is presented.

7.6.1 Preparing Data Sets from Sample and Measurement Results

Throughout Sect. 7.5, the terms *data* and *data point* are used interchangeably to describe the data set being evaluated. Typically, statistical evaluations are performed on results from sample or measurement data that have been collected from a systematic pattern of grid points across a survey unit. When the contamination is evenly dispersed within the survey unit, the contaminant is considered to be *homogeneously* mixed. Independent measurements from a homogeneous media can be evaluated directly when

* The term large survey unit is used here to mean areas that are larger than the maximum averaging area sizes required by the current DOE release criteria; i. e., 100 m² outdoors and 1 m² indoors.

performing statistical tests. In essence, each measurement or sample result is assumed to equal the activity level in the media surrounding the actual measurement or sample location and is considered to represent an area equal to the size of the sample/measurement grid block.

When contamination is primarily found as localized activity areas, i.e., hot spots, then the direct manipulation of independent measurements or samples collected from a systematic pattern is not likely to be appropriate unless the localized regions of contamination are properly represented. For these cases, the average contamination level for each grid block containing hot spots should be calculated as described in Sect. 7.5.3. The weighted average for each grid block will represent a *data point* in the statistical tests and will be representative of the amount of activity contained within it. As mentioned before, if the distribution of contamination within a sample/measurement grid block approximates a homogenous mixture, then a single sample/measurement should suffice as being representative of the remaining area— i.e., no weighted averaging should be necessary. For practical application, if a sample grid block contains a significant amount of contamination, but only within a sub-section of the whole, then a weighted average should be performed to determine the data value that is representative of that block. As a rule, higher frequencies of systematic samples will likely result in less need for performing averaging calculations since smaller regions are being represented by each systematic sample. Professional judgement will be required when determining whether a single sample or measurement provides an adequate representation of the surrounding media.

Illustration

To illustrate the above discussion, Fig. 7.1 shows an example site drawing with an overlay of a systematic sample grid. The solid dots show systematic sample locations and the dashed lines indicate the perimeter of the area being represented by the samples. In the absence of additional data, the single sample must be assumed to equal the activity for the entire block surrounding it. The diagram also shows three bounded areas of contamination containing activity levels significantly above the remaining area, i.e., hot spots. For the purpose of this illustration, assume that the elevated areas have been well bounded by field measurements and samples.

As can be seen, the systematic sample results for approximately seven of the blocks will give estimates of activity that are significantly less than the true amount within the respective grid area. Additionally, one of the sample results will bias the activity estimate for the block higher than the actual amount. Situations such as this dictate that weighted averages be calculated for the affected sample grid blocks, i.e., those that contain significant and non-homogeneous contamination. When performing subsequent statistical evaluations, these average values are used for the affected blocks instead of the single sample results. For the remaining grid areas, the single sample results should provide reasonable estimates for the entire block. The final data set will consist of a list of values, where each quantity reasonably represents

the average amount of activity contained within one of the sample grid areas. The use of this approach will assure that all activity, including that which is concentrated

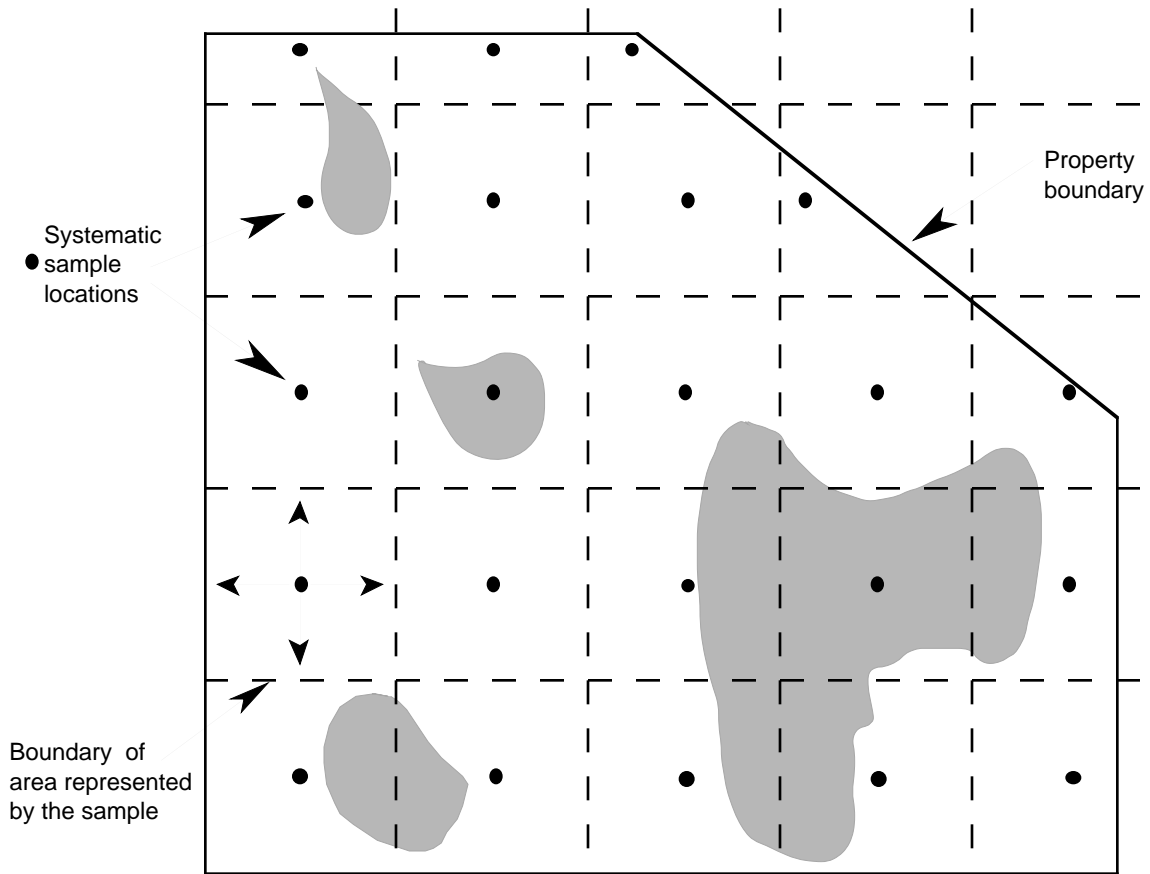


Fig. 7.1. Example site illustrating the necessity for weighted averaging of sample results.

in localized regions, is given proper weight when performing statistical comparisons with the data set.

7.6.2 Comparisons When Data are Normally Distributed and Background is a Known Constant

This section shows how to test if a survey unit meets a specified guideline value for the special situation where the data are normally (Gaussian) distributed and the background mean is known with no uncertainty. Section 7.6.3 provides a test that is applicable when the data are not normally distributed and the background mean is known with certainty. Appendix C provides two tests for when the background mean is not known with certainty: a test that requires normally distributed data, and a nonparametric test that can be used when data are not normally distributed. This latter nonparametric test is recommended for general use because fewer assumptions are required for it to give valid results.

Average levels, calculated following the procedures in Sect. 7.5.4, are compared with the guideline values and conditions. If the averages exceed the applicable guideline values and/or conditions, further remediation is required and follow-up measurements are performed to verify the effectiveness of the actions. After the averages satisfy the guideline values and conditions, the results are further evaluated to determine whether the data for each survey unit (i.e., group of contiguous grids or regions with the same classification of contamination potential) provide a 95% confidence level that the guidelines have been met.

The test is performed by calculating the average [Eq. (7.11)] and standard deviation of the data for a particular radiological parameter in each survey unit using all measurement locations. The standard deviation(s) of the mean is calculated by:

$$s = \sqrt{\frac{\sum_{i=1}^n (\bar{x} - x_i)^2}{n-1}} \quad (7.13)$$

The following equation (discussed in EPA 1989) can be used for testing data relative to a guideline value, at a desired level of confidence.

$$\mu_{\alpha} = \bar{x} + t_{1-\alpha, df} \frac{s}{\sqrt{n}} \quad (7.14)$$

where

$t_{1-\alpha, df}$ is the 95% confidence level obtained from Table 4.2 (Sect. 4): df (degrees of freedom) is $n-1$. α is the false positive probability, i.e. the probability that μ_{α} is less than the guideline value if the true mean activity level is equal to the guideline value,
 \bar{x} is the calculated mean from Eq. (7.13),

- s is the standard deviation from Eq. (7.14),
- n is the number of individual data points used to determine \bar{x} and s ,
- μ_{α} is the upper one-sided 95% confidence limit on the true mean for the survey unit.

The value of μ_{α} is compared to the guideline value; if μ_{α} is less than the guideline, the area being tested meets the guideline at a 95% confidence level. This means that the probability is less than 5% that the true mean activity exceeds the guideline value if μ_{α} is less than the guideline value.

Sample Calculation 1

Surface activity levels (dpm/100 cm²) for 35 systematic grid blocks in an affected (i.e., ≤ 1 m²) area are:

60*	5,400	7,830	10,400
120*	-120*	120*	-30*
-150 *	0*	30*	-90*
-270*	-150*	-210*	1,890
-60*	270*	1,170	-300*
90*	890	-120*	
3,000	180*	210*	
60*	-90*	30*	
210*	-30*	2,420	
-60*	330*	150*	

*These counts were less than the detection limit for the instrument, and negative values were less than instrument background.

All values were used in averaging (Sect. 7.5). Instrument background has already been subtracted for these surface activity measurements. When reported, actual values should be listed in parentheses following the detectable limit [e.g., <450 (-120)].

The parameters for this group of data are

$$\begin{aligned}
 t_{1-\alpha, df} &= 1.692 \text{ for 34 degrees of freedom (Sect. 4, Table 4.4)} \\
 \bar{x} &= 948 \\
 s &= 2354 \\
 \mu_a &= 948 + 1.692 \frac{2354}{\sqrt{35}} = 1621 \text{ dpm / 100 cm}^2
 \end{aligned}$$

The site-specific guideline value for the site is 5000 dpm/100 cm². Although 3 of the measurements exceed the average guideline value, it is assumed for the purposes of this example that the maximum level and the average for each averaging unit have been satisfied. Because μ_{α} is less than 5000 dpm/100 cm², the data for this survey unit satisfy the guideline at the 95% confidence level.

Sample Calculation 2

Concentrations of net (background subtracted) activity for 20 systematic soil sampling locations are

1.2 pCi/g	1.5 pCi/g
2.3 pCi/g	2.7 pCi/g
4.4 pCi/g	5.0 pCi/g
2.3 pCi/g	1.6 pCi/g
3.4 pCi/g	3.5 pCi/g
1.6 pCi/g	3.1 pCi/g
0.9 pCi/g	1.7 pCi/g
1.6 pCi/g	1.1 pCi/g
3.3 pCi/g	1.4 pCi/g
2.4 pCi/g	2.2 pCi/g

For the purposes of this example, it is assumed that these values accurately reflect the activity level for each respective grid blocks. The guideline value for the site is 4 pCi/g above background.

Although two of the samples contain activity levels above the average guideline value, for the purposes of this example it is assumed that the maximum guideline level for localized areas is not exceeded.

The mean and standard deviation for this group of data are:

$$\begin{aligned}\bar{x} &= 2.36 \text{ pCi/g [from Eq. (7.11)]} \\ s &= 1.12 \text{ pCi/g [from Eq. (7.12)]} \\ t_{1-\alpha, df} &= 1.729 \text{ for 19 degrees of freedom (Table 4.2, Sect. 4)} \\ \mu_{\alpha} &= 2.36 + 1.729 \frac{1.12}{\sqrt{20}} = 2.79 \text{ pCi/g}\end{aligned}$$

Comparison of μ_{α} (2.79 pCi/g) with the guideline value (4 pCi/g) indicates that the guideline has been satisfied at the desired level of confidence.

Areas for which μ_{α} is \leq the guideline values by this testing procedure are considered acceptable and no further survey actions are required. If the mean value exceeds the guideline value, the area is not acceptable and further cleanup is required. If the mean value is less than the guideline value, but the test of confidence is inconclusive (i.e., $\bar{x} < \text{guideline value} < \mu_{\alpha}$) either (1) further cleanup with follow-up measurements/sampling, or (2) additional measurements/sampling may be conducted.

7.6.3 Nonparametric Upper 95% Confidence Limit Test

The upper 95% confidence limit on the mean was used in Sect. 7.6.2 to test if the survey unit meets the guideline limit. That test requires the survey-unit data to

have a normal distribution and for the background mean to be known with no uncertainty. (See Appendix B for examples of tests when the background value is not precisely known). When the data are not normally distributed, the nonparametric (distribution-free) upper one-sided 95% confidence limit on the median may be used. However, this test also requires that the background mean be known with no uncertainty. The test is conducted as follows (Gilbert 1987, p. 173):

1. Order the n net (background-corrected) measurements in the survey unit from smallest to largest.
2. If $n \leq 150$, find the value u in Table 7.2 that corresponds to n . (The meaning of it is explained below). If $n \geq 150$, find u in the table on the bottom half of p. 104 of Geigy (1982), or compute u as follows:

$$u = (n + 1 + Z_{1-\alpha} n^{1/2}) / 2$$

where $Z_{1-\alpha}$ is the $(1 - \alpha)$ th percentile of the standard normal distribution. When $\alpha = 0.05$, $Z_{1-\alpha}$ become $Z_{0.95}$, which is equal to 1.645.

3. Find the u th largest measurement (starting from the smallest measurement) in the ordered list of measurements (from Step 1). This measurement is the upper 95% confidence limit on the true median of background-corrected measurements for the survey-unit.
4. If the u th largest value is less than the guideline value, then the survey unit meets the guideline at the 95% confidence level.

Example 1

The test is illustrated using the $n = 35$ data for “Sample Calculation 1” in Sect. 7.6.2. From Table 7.2, $u = 23$ when $n = 35$. From the ordered list of the 35 data points, the 23rd largest measurement [counting from the smallest measurement (–300) upward] is 150 dpm/100 cm². The specified guideline value was 5000 dpm/100 cm². Therefore, the survey unit satisfies the guideline because $150 < 5000$. This is the same conclusion that was obtained using the test in Sect. 7.6.2 based on Eq. (7.14).

Example 2

The $n = 20$ data for “Sample Calculation 2” in Sect. 7.6.2 are used. From Table 7.2, $u = 15$ when $n = 20$. The 15th largest measurement in the data set is 3.1 pCi/g. The guideline value is 4 pCi/g. Therefore, the survey unit satisfies the guideline because $3.1 < 4$.

Table 7.2. Factors u for conducting the nonparametric 95% upper confidence limit test (Sect. 7.5.6) for number of samples (n) from 1 to 150^a

n	u	n	u	n	u	n	u	n	u	n	u
1	-	26	18	51	32	76	46	101	60	126	73
2	-	27	19	52	33	77	47	102	60	127	74
3	-	28	19	53	33	78	47	103	61	128	74
4	-	29	20	54	34	79	48	104	61	129	75
5	5	30	20	55	35	80	48	105	62	130	75
6	6	31	21	56	35	81	49	106	62	131	76
7	7	32	22	57	36	82	49	107	63	132	76
8	7	33	22	58	36	83	50	108	64	133	77
9	8	34	23	59	37	84	51	109	64	134	78
10	9	35	23	60	37	85	51	110	65	135	78
11	9	36	24	61	38	86	52	111	65	136	79
12	10	37	24	62	38	87	52	112	66	137	79
13	10	38	25	63	39	88	53	113	66	138	80
14	11	39	26	64	40	89	53	114	67	139	80
15	12	40	26	65	40	90	54	115	67	140	81
16	12	41	27	66	41	91	54	116	68	141	81
17	13	42	27	67	41	92	55	117	68	142	82
18	13	43	28	68	42	93	55	118	69	143	82
19	14	44	28	69	42	94	56	119	69	144	83
20	15	45	29	70	43	95	57	120	70	145	83
21	15	46	30	71	43	96	57	121	71	146	84
22	16	47	30	72	44	97	58	122	71	147	84
23	16	48	31	73	45	98	58	123	72	148	85
24	17	49	31	74	45	99	59	124	72	149	86
25	18	50	32	75	46	100	59	125	73	150	86

^aFrom Geigy 1982, p. 104.

7.6.4 Comparing Survey-Unit Data with Background Data

This guidance report focuses on the question of whether background-corrected measurements exceed derived (fixed) levels such as release guidelines, derived limits, or standards. However, sometimes there is a need to know if concentrations in a survey unit are really greater than those in background. For example, this question may be evaluated for the purpose of identifying potential contaminants of concern. Gilbert and Simpson (December 1992) discuss and illustrate the nonparametric Wilcoxon Rank Sum test and the Quantile test for this purpose. Hardin and Gilbert (December 1993) evaluate the performance of these and other tests for the background-comparison case. They conclude that the Wilcoxon Rank Sum test is the best overall performer among the tests evaluated unless only a very small portion of the survey-unit is contaminated to high levels. In that case, the Quantile test is preferred. (NOTE: The nonparametric test shown in Appendix B is closely related to the Wilcoxon Rank Sum test.)

7.7 EVALUATING RESULTS RELATIVE TO DATA QUALITY OBJECTIVES

The concept of Data Quality Objectives (DQOs) and guidance in establishing appropriate values or criteria for the DQOs was described in Sect. 1.3. At completion of the survey, the overall performance, relative to satisfying the DQOs should be determined. Each indicator of data quality identified in the survey plan should be evaluated, either qualitatively or quantitatively, following the approach described in the plan. Comparison of the performance to the respective DQO should be made and the overall evaluation discussed in the survey report. The end use of the DQO evaluation in determining acceptability of the survey results is a subjective decision and requires consideration of all aspects of the procedures and findings.